

# Grunnlag for fastsettelse av grenseverdi

Grunnlagsdokument for bisfenol A  
(C<sub>15</sub>H<sub>16</sub>O<sub>2</sub>)



Kommisjonsdirektiv 2017/164/EU

Tittel: Grunnlag for fastsettelse av grenseverdi.  
Grunnlagsdokument for bisfenol A (C<sub>15</sub>H<sub>16</sub>O<sub>2</sub>).

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Denne rapporten omhandler det toksikologiske grunnlaget og vurderinger, samt tekniske og økonomiske hensyn for fastsettelse av grenseverdi for bisfenol A (C<sub>15</sub>H<sub>16</sub>O<sub>2</sub>).



# Innhold

Innhold	3
Forord	4
Innledning	5
1. Stoffets identitet	5
2. Fysikalske og kjemiske data	5
2.1 Forekomst og bruk	6
3. Grenseverdier	6
3.1. Nåværende grenseverdi	6
3.2. Grenseverdi fra EU	6
3.3. Grenseverdier fra andre land og organisasjoner	6
3.4. Stoffets klassifisering	7
3.5. Biologisk overvåking	8
4. Toksikologiske data og helseeffekter	8
4.1. Anbefaling fra SCOEL	8
4.2. Kommentarer fra TEAN	9
5. Bruk og eksponering	10
5.1. Opplysning fra Produktregistret	10
5.2. Eksponering og måledokumentasjon	11
5.2.1. EXPO- data	11
5.2.2. Prøvetakings- og analysemetode	11
6. Vurdering	11
7. Konklusjon med forslag til ny grenseverdi	13
8. Ny grenseverdi	13
9. Referanse	14
10. Vedlegg: SCOEL	15



## Forord

Grunnlagsdokumenter for fastsettelse av grenseverdier utarbeides av Arbeidstilsynet i samarbeid med Statens arbeidsmiljøinstitutt (STAMI) og partene i arbeidslivet (Næringslivets hovedorganisasjon/Norsk Industri og Landsorganisasjonen i Norge) i henhold til *Strategi for utarbeidelse og fastsettelse av grenseverdier for forurensninger i arbeidsatmosfæren*. Dette dokumentet er utarbeidet ved implementering av kommisjonsdirektiv 2017/164/EU fastsatt 31. januar 2017.

EU-rådets direktiv 98/24/EC (Vern av helse og sikkerhet til arbeidstakere mot risiko i forbindelse med kjemiske agenser på arbeidsplassen) av 7. april 1998 stiller krav om at EU-kommisjonen skal legge frem forslag til indikative grenseverdier for eksponering av visse kjemikalier som medlemslandene må innføre på nasjonalt nivå. De nasjonale grenseverdiene kan være høyere enn de som står oppført i direktivet, dersom et medlemsland mener at det er nødvendig av tekniske og/eller økonomiske hensyn, men landene bør nærme seg den indikative grenseverdien. Direktivet stiller krav om at indikative grenseverdier vedtas gjennom kommisjonsdirektiv.

I Norge ble de indikative grenseverdiene innført som veiledende administrative normer. Da nye Arbeidsmiljøforskrifter trådte i kraft 1.1.2013 ble de veiledende administrative normene forskriftsfestet i forskrift om tiltaks- og grenseverdier og fikk betegnelsen tiltaksverdier. I 2015 ble begrepet «grenseverdi» for kjemikalier presisert og begrepet «tiltaksverdi» for kjemikalier ble opphevet i forskrift om tiltaks- og grenseverdier. I vedlegg 1 til forskriften ble det innført en tydeliggjøring av anmerkningene.

Arbeidstilsynet har ansvaret for revisjonsprosessen og utarbeidelse av grunnlagsdokumenter for stoffene som blir vurdert. Det toksikologiske grunnlaget for stoffene i denne revisjonen baserer seg i hovedsak på kriteriedokumenter fra EUs vitenskapskomité for fastsettelse av grenseverdier, Scientific Committee for Occupational Exposure Limits (SCOEL). SCOEL utarbeider de vitenskapelige vurderingene som danner grunnlaget for anbefalinger til helsebaserte grenseverdier, og disse legges fram for kommisjonen.

Statens arbeidsmiljøinstitutt (STAMI) ved Toksikologisk ekspertgruppe for administrative normer (TEAN) bidrar med faglige vurderinger i dette arbeidet. TEAN vurderer og evaluerer de aktuelle SCOEL dokumentene, presiserer kritiske effekter og vurderer behov for korttidsverdier ut i fra den foreliggende dokumentasjonen. Videre søker og evaluerer TEAN nyere litteratur etter utgivelsen av dokumentet. TEAN bruker kriteriene gitt i SCOEL's metodedokument, "Methodology for the derivation of occupational exposure limits: Key documentation (version 7, June 2013)". Dette er inkludert i TEANs Metodedokument del B (Prosedyre for utarbeidelse av toksikologiske vurderinger for stoffer som skal implementeres i det norske regelverket for grenseverdier etter direktiv fra EU-kommisjonen) utarbeidet for denne revisjonen.

Informasjon om bruk og eksponering i Norge innhentes fra Produktregisteret, EXPO databasen ved STAMI og eventuelle tilgjengelige måledata fra virksomheter/næringer. Beslutningsprosessen skjer gjennom drøftingsmøter der Arbeidstilsynet, Næringslivets hovedorganisasjon/Norsk Industri og Landsorganisasjonen i Norge deltar, samt orienteringsmøter og offentlig høring. Konklusjonene fra høringen med forskriftsendringer og nye grenseverdier forelegges Arbeids- og sosialdepartementet som tar den endelige beslutningen.



# Innledning

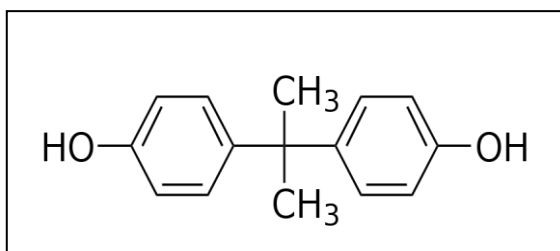
Dette grunnlagsdokumentet omhandler vurderingsgrunnlaget for fastsettelse av grenseverdi for bisfenol A. Innholdet bygger spesielt på anbefalinger fra Scientific Committee on Occupational Exposure Limits (SCOEL) i EU for bisfenol A (vedlegg 1), samt vurderinger og kommentarer fra Toksikologisk Ekspertgruppe for Administrative Normer (TEAN).

## 1. Stoffets identitet

Bisfenol A og dets molekylformel, synonymer av stoffets navn, stoffets identifikasjonsnummer i Chemical Abstract Service (CAS-nr.), European Inventory of Existing Commercial Chemical Substances (EINECS-nr. el. EC-nr.) og indekseringsnummer (Indeks-nr.) er gitt i tabell 1. Strukturformel av bisfenol A er vist i figur 1.

**Tabell 1.** Stoffets navn og identitet.

<b>Navn</b>	Bisfenol A
<b>Molekylformel</b>	C <sub>15</sub> H <sub>16</sub> O <sub>2</sub>
<b>Synonymer</b>	4,4'-Isopropylidendifenol, 2,2-Bis (4 hydroxy-fenyl) propan
<b>CAS-nr.</b>	80-05-7
<b>EC-nr.</b>	201-245-8
<b>Indeks-nr.</b>	604-030-00-0



**Figur 1.** Strukturformel av bisfenol A (Wikipedia, [https://sv.wikipedia.org/wiki/Bisfenol\\_A](https://sv.wikipedia.org/wiki/Bisfenol_A)).

## 2. Fysikalske og kjemiske data

Det vises til tabell 2 for fysikalske og kjemiske data for bisfenol A.

**Tabell 2.** Tabellen nedenfor viser fysikalske og kjemiske data for bisfenol A.

<b>Kjemisk formel</b>	C <sub>15</sub> H <sub>16</sub> O <sub>2</sub>
<b>Molekylvekt</b>	228,29
<b>Fysisk tilstand</b>	Hvite til lysebrune flak eller pulver
<b>Smeltepunkt (°C)</b>	154-157*
<b>Kokepunkt (°C)</b>	360
<b>Flammepunkt (°C)</b>	207**
<b>Selvantennelsestemperatur (°C):</b>	532
<b>Eksplosjonsgrenser (g/l)</b>	Minimum eksplosiv konsentrasjon 0,012 (O <sub>2</sub> > 5%)
<b>Løselighet i vann (25 °C)</b>	0,298 g/l
<b>Løselighet i eter og alkohol (20 °C)</b>	-
<b>Damptrykk ved 20 °C (kPa)</b>	4.12x10 <sup>-10</sup> ***
<b>Omregningsfaktor (20 °C, 101 kPa)</b>	1 ppm = 9,50 mg/m <sup>3</sup> , 1 mg/m <sup>3</sup> = 0,105 ppm

\*Smeltepunkt er avhengig av produksjonsprosess, se <https://echa.europa.eu/registration-dossier/-/registered-dossier/15752/4/3/?documentUUID=b7529a2c-b2b3-47e1-ba5a-b0ac1fa08179>

\*\*<https://pubchem.ncbi.nlm.nih.gov/compound/6517#section=Flash-Point>

\*\*\*<https://echa.europa.eu/registration-dossier/-/registered-dossier/15752/4/7>, EC 2003 bruker annen verdi som ikke er riktig, jf The EU risk assessment report (European Chemicals Bureau, 2003).

## 2.1 Forekomst og bruk

Bisfenol A framstilles av fenol og aceton i en syre- eller alkalie-katalysert kondensasjonsreaksjon.

Bisfenol A (heretter også kalt BPA) brukes i EU hovedsakelig (over 95 %) i produksjon av polykarbonat-plast og epoksyharpikser. Annen bruk er som flammehemmende middel og ved framstilling av polyester, polyakrylat, polyeterimid og polysulphone harpikser. Produksjonen av BPA i EU var 1 438 tusen tonn i 2008 og anses ikke å ha økt særlig de siste årene. Se forøvrig avsnitt 5.1. om bruk og forekomst av bisfenol A i Norge.

## 3. Grenseverdier

### 3.1. Nåværende grenseverdi

Nåværende grenseverdi i Norge for bisfenol A er: 10 mg/m<sup>3</sup> med anmerkning ARE fastsatt i 2011.

### 3.2. Grenseverdi fra EU

Den europeiske vitenskapskomiteen, SCOEL foreslår for bisfenol A i sitt kriteriedokument fra 2014: IOELV (Indicative Occupational Exposure Limit Value): 2 mg/m<sup>3</sup> (inhalerbar)

### 3.3. Grenseverdier fra andre land og organisasjoner

**Tabell 3.** Tabellen nedenfor viser grenseverdier for bisfenol A fra andre land og organisasjoner.



Land Organisasjon	Grenseverdi (8 timer) mg/m <sup>3</sup>	Korttidsverdi (15 min)	Anmerkning Kommentar
Sverige <sup>1</sup>	2	-	R, 2, (Inhalerbar)
Danmark <sup>2</sup>	3	-	E (EF-grænseverdi)
Finland <sup>3</sup>	5	-	-
Storbritannia <sup>4</sup>	10	-	-
Nederland <sup>5</sup>	-	-	-
ACGIH, USA <sup>6</sup>	-	-	-
NIOSH, USA <sup>6</sup>	-	-	-
Tyskland, MAK <sup>6</sup>	5	-	I (Inhalerbar, 1), C (reproduksjonsskadelig, gruppe C), SP (sensibilisering ved foto-kontakt)
Tyskland, Myndighetene <sup>7</sup>	5	-	Y (ikke fare for skade på foster dersom grenseverdi overholdes)

<sup>1</sup> Arbetsmiljöverkets Hygieniska gränsvärden AFS 2015:7,

<https://www.av.se/globalassets/filer/publikationer/foreskrifter/hygieniska-gransvarden-afs-2015-7.pdf>.

<sup>2</sup> At-vejledning, stoffer og materialer - C.0.1, 2007, <https://arbejdstilsynet.dk/da/regler/at-vejledninger/g/c-0-1-graensevaerdi-for-stoffer-og-mat>.

<sup>3</sup> Social og hälsovårdsministeriet, HTP-värden, Koncentrationer som befunnits skadliga, Helsingfors, 2016, [http://julkaisut.valtioneuvosto.fi/bitstream/handle/10024/79110/STM\\_9\\_2016\\_HTP-yarden\\_2016\\_Ruotsi\\_22122016\\_NETTI.pdf](http://julkaisut.valtioneuvosto.fi/bitstream/handle/10024/79110/STM_9_2016_HTP-yarden_2016_Ruotsi_22122016_NETTI.pdf).

<sup>4</sup> EH40 andre utgave, 2013, <http://www.hse.gov.uk/pubns/priced/eh40.pdf>

<sup>5</sup> [http://www.ser.nl/en/oel\\_database.aspx](http://www.ser.nl/en/oel_database.aspx);

<http://www.ser.nl/en/grenswaarden/2%20butyne%201%204%20diol.aspx>

<sup>6</sup> Guide to occupational exposure values compiled by ACGIH, 2017.

<sup>7</sup> Baua, TRGS 900, oppdatert 2016, [https://www.baua.de/DE/Angebote/Rechtstexte-und-Technische-Regeln/Regelwerk/TRGS/pdf/TRGS-900.pdf;jsessionid=439FFF321DF2323E60F868CD08E9CD3A.s1t2?\\_blob=publicationFile&v=2](https://www.baua.de/DE/Angebote/Rechtstexte-und-Technische-Regeln/Regelwerk/TRGS/pdf/TRGS-900.pdf;jsessionid=439FFF321DF2323E60F868CD08E9CD3A.s1t2?_blob=publicationFile&v=2)

### 3.4. Stoffets klassifisering

Bisfenol A er i henhold til CLP Annex VI, tabell 3.1 (Liste over harmonisert klassifisering og merking av farlige kjemikalier) klassifisert og merket i ulike fareklasser, med faresetninger og koder, som gitt i tabell 4 nedenfor.

**Tabell 4.** Fareklasser, farekategori med forkortelse, merkekoder og faresetninger for bisfenol A<sup>1</sup>

Fareklasse, Farekategori, Forkortelse	Merkekode	Faresetning
Alvorlig øyeskade/øyeirritasjon, kategori 1	H318	Gir alvorlig øyeskade.
Sensibiliserende ved innånding eller hudkontakt Hudsensibilisering, Kategori 1, Underkategori 1A, Underkategori 1B (Skin Sens., 1/1A/1B).	H317	Kan utløse en allergisk hudreaksjon
Spesifikk målorgantoksisitet – enkelt eksponering, kategori 3 (STOT SE 3)	H335	Kan forårsake irritasjon av luftveiene.
Reproduksjonstoksisitet, Kategori 1B (Repr1B)	H360F*	Kan skade forplantningsevnen.



<sup>1</sup> CLP ((Forordning (EC) Nr. 1272/2008), <http://www.miljodirektoratet.no/Documents/publikasjoner/M259/M259.pdf>  
<https://echa.europa.eu/information-on-chemicals/cl-inventory-database>. \*F=Fruktbarhet

### **3.5. Biologisk overvåking**

For å beskrive eksponering for forurensning i luften på arbeidsplassen kan man anvende konsentrasjonen av forurensningen i arbeidstakerens urin, blod eller utåndingsluft, eller annen respons på eksponeringen i kroppen. EU har satt verdier for dette kalt biologisk grenseverdi (BLV).

SCOEL fremmer ikke forslag til BLV for bisfenol A, men foreslår en veiledende verdi ved måling av bisfenol A i urin (Benchmark guidance value, BGV) for stoffet lik 7 µl/l (urinary total bisphenol A).

## **4. Toksikologiske data og helseeffekter**

### **4.1. Anbefaling fra SCOEL**

EUs vitenskapskomite (SCOEL) har utarbeidet kriteriedokumentasjon for bisfenol A datert juni 2014 hvor de anbefaler en grenseverdi for stoffet lik 2 mg/m<sup>3</sup> (inhalerbar), se vedlegg.

SCOEL har gitt sin anbefaling til grenseverdi for bisfenol A på grunnlag av en inhalasjonsstudie hvor det ble fastsatt en NOAEC på 10 mg/m<sup>3</sup> (Nitschke 1988). NOAEL ble satt til 10 mg/m<sup>3</sup>, som er grunnlaget for deres anbefaling til OEL (grenseverdi) på 2 mg/m<sup>3</sup>. Den kritiske effekt i denne studien var irritasjon av øvre luftveier. NOAEC ble dividert med en usikkerhetsfaktor på 3 som resulterte i en anbefalt OEL på 3 mg/m<sup>3</sup>, rundet ned til 2 mg/m<sup>3</sup>. Langtids systemiske effekter på lever and nyrer ble ikke undersøkt i denne inhalasjonsstudien.

På grunnlag av orale studier har SCOEL angitt en BMDL10 på ~3.5 mg/kg bw (EFSA 2014) og en NOAEL på 5 mg/kg bw for effekter på lever og nyre i repeterte oralstudier (Stump et al 2010, Tyl et al 2002 and 2008). Dette ved eksponeringsnivåer på 34 and 49 mg/m<sup>3</sup>, sikkerhetsmargin 17–25. Langtidsstudiene viste imidlertid milde effekter, selv ved de høyeste dosenivå (50 mg/kg og 500/600 mg/kg kroppsvekt, Tyl et al 2002 and 2008). Disse data anses derved ikke å kunne vektlegges i anbefaling av OEL.

SCOEL angir at det ikke er toksikologisk grunnlag for å foreslå korttidsverdi for bisfenol A (STEL).

SCOEL angir også at det er begrensede data for å fastsette en biologisk grenseverdi BLV. I den generelle befolkningen er nivå av BPA i urin vanligvis under 7 µg/l (95-prosentil basert på tyske og kanadiske studier). Det er imidlertid stor usikkerhet i beregningene på grunn stor variasjon i nivåer over en dag, på grunn av den korte halveringstiden til BPA. Det er også begrensede data om nedbrytingen av BPA etter inhalasjon eller hudpåvirkning, og få data finnes om oral eksponering. Derfor anbefales ingen BLV ut over en BGV (veiledende verdi) på 7 µg/l som kan benyttes for å påvise om eksponering har forekommet. Denne verdi er satt av det tyske miljøinstituttet (UBA, 2012).

SCOEL anbefaler ingen anmerkning om hudopptak fordi hudopptaket ved eksponering på nivå med den foreslåtte grenseverdi vil bidra lite til opptak av bisfenol A i kroppen.





## 4.2. Kommentarer fra TEAN

SCOEL sitt grunnlagsdokument er fra 2014 og baserer seg i stor grad på gjennomganger gjort av EFSA (2014), WHO (2011) og EU-rapporter (2003 og 2008). I de siste årene har det vært stor forskningsaktivitet rundt bisfenol A og det finnes derfor mye nye data. En hovedgrunn er at stoffet er mistenkt å kunne påvirke spesifikke hormonsystemer. Siden hormoner er svært potente og har viktige funksjoner ved svært lave konsentrasjoner, er det mistanke om at stoffer som påvirker hormonell aktivitet også kan ha skadelige effekter ved lave eksponeringsnivåer.

Nøkkelstudiene i SCOEL-rapporten er den samme som de brukte i sin bisfenol A rapport fra 2004: En 13 ukers inhalasjonsstudie med rotter hvor man fant svak irritasjon i øvre luftveier ved konsentrasjoner på 50 mg/m<sup>3</sup> og høyere (Nitschke, 1988). NOAEL ble satt til 10 mg/m<sup>3</sup>, som er grunnlaget for å sette en OEL på 2 mg/m<sup>3</sup>. Dette tilsvarer et daglig inntak på 0.29 mg/kg kroppsvekt(kv). SCOEL diskuterer også en multigenerasjonsstudie med mus og rotter hvor bisfenol A ble gitt oralt. Dette er nøkkelstudien i EFSA-rapporten fra 2014 (endelig versjon utgitt i 2015) hvor de kritiske effektene var endringer av organvekter for nyre og lever. Basert på disse studiene ble NOAEL av EFSA satt til 5mg/kg kv/dag.

I EFSA har kritisk effekt og NOAEL vært uendret siden deres rapport på bisfenol A fra 2006, mens TDI-verdien (tolerabelt daglig inntak) de har satt, er senket fra 50 ug/kg kv i 2006 til 4 ug/kg kv i 2015-rapporten. EFSA sier at den satte TDI i 2015 er midlertidig (kalt t-TDI) og hvor de har tatt hensyn til stor usikkerhet rundt helseeffekter av inntak av bisfenol A. Den midlertidige verdien er satt i påvente av at US-NTP kommer med sin rapport fra forskningsprogrammet CLARITY-BPA. Her er det et hovedfokus på skadelige effekter knyttet til hormonforstyrrende mekanismer. I de siste årene er det publisert flere studier som kan tyde på at bisfenol A har skadelige effekter bl.a. på reproduksjon, immunsystem, fettmetabolisme og utvikling av nervesystem.

ECHA har fra 12/1-2017 plassert bisfenol A på kandidatlisten for SVHC stoffer. Begrunnelsen er at stoffet kan være toksisk for reproduksjon og at det kan ha hormonforstyrrende egenskaper. Stoffet har på dette grunnlag harmonisert klassifisering som Reprotoksisk kategori 1B.

TEAN mener at det foreliggende kriteriedokument fra SCOEL i liten grad har vurdert mulige helseeffekter knyttet til hormonforstyrrende mekanismer av bisfenol A. Slike studier har blitt langt flere etter 2014, og det er nødvendig å se nærmere på sammenhengen mellom slike helseeffekter og eksponeringsdoser. Den foreslåtte grenseverdien bør derfor ansees som midlertidig, analogt med hvordan EFSA har vurdert usikkerhet i kunnskapsnivået. Oppsummeringen av resultater fra CLARITY-BPA programmet i USA vil trolig bidra til å gjøre risikovurdering av bisfenol A mer sikker. Denne er annonsert å komme rundt årsskifte 2017/2018.

### Biologisk grenseverdi

BPA metaboliseres til konjugater med glukuronsyre og sulfat. Hoveddelen av disse utskilles i urinen, nær 100% i løpet av et døgn. SCOEL foreslår en biologisk veiledende verdi på 7 ug/l urin. I generell befolkning ligger urin BPA vanligvis lavere enn denne. Den foreslåtte OEL (2 mg/m<sup>3</sup>) er beregnet å tilsvare en urinkonsentrasjon på 11.8 mg/l. BPA har en kort halveringstid, så det er store usikkerheter i disse beregningene. SCOEL foreslår derfor ingen biologisk grenseverdi.

En rekke studier viser urinmetabolitter av bisfenol A som kan knyttes til yrkeseksponering. Det er kjent at produksjon og håndtering av varmesensitivt papir (som bl.a. brukes i kassalapper) kan gi betydelig hudopptak av bisfenol A. I en studie av 90 kassabøtjenter var medianverdiene ca 9 ug/l urin og signifikant høyere enn hos 40 kontroller (3.5 ug/l).



SCOEL foreslår ingen hudenmerkning, trolig fordi hudopptaket blir lite sammenlignet med inhalasjonsopptaket når luftkonsentrasjonen er nær den foreslåtte grenseverdien.

## 5. Bruk og eksponering

Nedenfor er angitt bruken av bisfenol A i Norge og bransjer hvor eksponering kan forekomme.

### 5.1. Opplysning fra Produktregisteret

Data fra Produktregisterets er innhentet oktober 2016, og inneholder opplysninger om mengde og bruk av bisfenol A i 64 deklareringspliktige produkter. Netto maksimal mengde av bisfenol A i disse produktene utgjør 12, 97 tonn.

Bisfenol A inngår særlig i produksjon av maling og lakk, ved produksjon, støping og overflatebehandling av metaller, i bygging, vedlikehold og reparasjon av skip/båter, innen bygge og anleggsvirksomhet, og særlig ved gulvlegging (epoksy).

Det henvises til tabell 5 for detaljert oversikt over bransjebeskrivelser med tilhørende bransjekode for de produkter det kan rapporteres på, og total mengde utgjør 6,216 tonn.

**Tabell 5.** Oversikt over bransjer hvor bisfenol A benyttes og mengde forbruk i tonn.

Bransjekode	Beskrivelse	Maksimal mengde (tonn)
20.3	Produksjon av maling og lakk, trykkfarger og tetningsmidler	1,276
24	Produksjon av metaller	0,506
24.5	Støping av metaller og lettmetall	1,188
25.61	Overflatebehandling av metaller	2,543
30.11	Bygging av skip og flytende materiell	0,506
32	Annen industriproduksjon	0,465
33.15	Reparasjon og vedlikehold av skip og båter	0,459
43	Spesialisert bygge- og anleggsvirksomhet, herunder også malerarbeid (43.341)	0,935
43.33	Gulvlegging og tapetsering	0,736
46,75	Engroshandel med kjemiske produkter	0,572

Opplysninger om produkttypekode, produkttype og maksimal mengde (over 0,4 tonn) er gitt i tabell 6 nedenfor (i alt over 8 tonn).

Nedenfor viser data over typer produkter og mengde bisfenol A.

**Tabell 6.** Oversikt over sentrale produkttyper som inneholder bisfenol A og maksimale mengder.



Produkttypekode	Produkttype	Maksimal mengde (tonn)
H15400	Plastherdere	0,429
H15500	Andre herdere	2,442
M05543	Maling og lakk pulverlakk dekorativ/beskyttelse, industrielt bruk	2,532
M05631 og M05633	Maling og lakk herdere til maling og lakk, aktiv korrosjonsbeskyttende effekt	1,967
M05643	Maling og lakk herdere til maling og lakk, dekorativ/beskyttelse, industriell og annen bruk	0,729

På grunn av sikkerhetsbestemmelsene i Produktregisteret kan vi ikke gi eksakte opplysninger ut over informasjon gitt i tabellene 5 og 6.

## 5.2. Eksponering og måledokumentasjon

Eksponeringene for bisfenol A i arbeidslivet i Norge er ikke undersøkt, og er således ukjent.

### 5.2.1. EXPO- data

Det er ikke registrert målinger av bisfenol A i STAMIs eksponeringsdatabase EXPO.

### 5.2.2. Prøvetakings- og analysemetode

I tabell 7 er anbefalte metoder for prøvetaking og analyser av bisfenol A presentert.

**Tabell 7.** Anbefalte metoder for prøvetaking og analyse av bisfenol A.

Prøvetakingsmetode	Analysemetode	Referanse
Glassfiberfilter	Desorpsjon m/HPLC/UV	OSHA-metode 1018 <sup>1</sup>

<sup>1</sup> [www.osha.gov/dts/sltc/methods/toc.html](http://www.osha.gov/dts/sltc/methods/toc.html)

Spesiell metode for prøvetaking av bisfenol A er beskrevet av Peltonen K, Pukkila J: Determination of bisphenol A in air by high performance liquid chromatography with electrochemical detection. J. Chrom. 1988; 439: 375 – 380.

## 6. Vurdering

Eksponering for bisfenol A i arbeidslivet i Norge er ikke undersøkt, og nivå av eksponering er således ukjent. Data fra Produktregisteret viser imidlertid at rundt 13 tonn bisfenol A kan inngå i totalt 64 produkter brukt i Norge, særlig ved maling, lakkering og overflatebehandling innen industri, og bygg og anlegg. Bisfenol A inngår som stoff i herdere og særlig i epoxyprodukter. Rundt 8 tonn av produktene brukes innen disse bransjene (se tabell 5), hvorav rundt 5,5 tonn utgjør herdere. En begrenset andel av denne mengde vil gi grunnlag for eksponering for arbeidstakere, og sammenlignet med annen bruk av kjemikalier i Norge er mengdene, og derved konsekvensene av lavere eksponeringsgrense, begrenset for berørte bransjer.



Anbefalingen fra SCOEL til grenseverdi er basert på nøkkelstudien i EFSA sin rapport fra 2014 (endelig versjon utgitt i 2015). Kritiske effekter er irritasjon i øvre luftveier ved konsentrasjoner på 50 mg/m<sup>3</sup> og høyere (Nitschke 1988). NOAEL ble satt til 10 mg/m<sup>3</sup>, som er grunnlaget for å sette en OEL på 2 mg/m<sup>3</sup>.

I tillegg er det referert til orale studier hvor de kritiske effektene var endringer av organvekter for nyre og lever (multigenerasjonsstudie med mus og rotter hvor bisfenol A ble gitt oralt), men disse har ikke vært relevante for bruk i vurdering av eksponeringsgrenser for bisfenol A.

Det foreligger begrenset med studier av effekter av eksponering for BPA ved innånding ved yrkesmessig bruk, som er den sentrale eksponeringsvei. Det er også manglende data for effekter på hud ved yrkeseksponering til å kunne konkludere spesielt om hudsensibiliserende effekter, men dette kan ikke utelukkes.

Selv om ikke mekanismene rundt mulig hudsensibiliserende fare knyttet til bisfenol A er kjent, er det studier som viser fotosensibiliserende egenskaper ved bisfenol A ved samtidig UV-stråling (Allen and Kaidbey 1979, Gerberick and Ryan 1990, Maguire 1988), og dette er det grunnlag for å ta med i vurderingen av helsefaren ved hudpåvirkning. Det er påvist tilfeller av hudskader av gjentatt eksponering av bisfenol A (jf vedlegg).

SCOEL vurderer imidlertid at anmerkning om hudopptak (Sen, eller H) ikke er relevant fordi hudopptaket ved eksponering på nivå med den foreslåtte grenseverdi sannsynligvis bidrar lite til opptak av bisfenol A i kroppen.

TEAN mener at det foreliggende kriteriedokument fra SCOEL i liten grad har vurdert mulige helseeffekter knyttet til hormonforstyrrende mekanismer av bisfenol A som er funnet i studier etter 2014, og at den foreslåtte grenseverdien derfor bør vurderes på nytt i lys av studier av sammenhengen mellom slike helseeffekter og eksponeringsdoser i dag og i framtiden. Siden hormoner er svært potente og har viktige funksjoner ved svært lave konsentrasjoner, er det mistanke om at stoffer som påvirker hormonell aktivitet også kan ha skadelige effekter ved lave eksponeringsnivåer.

TEAN fremhever også at det de siste årene er publisert flere studier som kan tyde på at bisfenol A har skadelige effekter bl.a. på reproduksjon, immunsystem, fettmetabolisme og utvikling av nervesystem, og at den anbefalte grenseverdi fra SCOEL på 2 mg/m<sup>3</sup> må vurderes på nytt når mer sikre data om risikoen med bisfenol A kan komme i forbindelse med det pågående CLARITY-BPA programmet i USA som vil foreligge i 2018.

Stoffet er også i CLP klassifisert som øyeskadelig, sensibiliserende ved innånding eller hudkontakt, spesifikk målorgantoksisitet (enkelteksponering, kategori 3) og reproduksjonsskadelig (Repr 1B).

SCOEL angir at det er begrensede data for å fastsette en biologisk grenseverdi BLV. Derfor anbefales av dem ingen BLV ut over en BGV (veiledende verdi) på 7 µg/l som kan benyttes for å vurdere om eksponering har forekommet.

SCOEL angir også at det ikke er toksikologisk grunnlag for å fastsette korttidsverdi for bisfenol A.

## 7. Konklusjon med forslag til ny grenseverdi

På bakgrunn av den foreliggende dokumentasjon og en avveining mellom de toksikologiske dataene og data om bruk av produkter i Norge som inneholder bisfenol A (dvs. tekniske og økonomiske hensyn), foreslås at dagens grenseverdi reduseres til nivå anbefalt av SCOEL, og å beholde anmerkningene allergifremkallende (A) og reproduksjonsskadelig (R). Grenseverdien foreslås også ut fra hensynet til at det mangler kunnskap om potensielle effekter av BPA på reproduksjon, immunsystem, fettmetabolisme og utvikling av nervesystem.

Forslag til ny grenseverdi, korttidsverdi og anmerkning:

**Grenseverdi (8-timers TWA): 2 mg/m<sup>3</sup> (inhalerbar)**

**Anmerkning: A (Allergifremkallende), R (reproduksjonsskadelig) og E (EU har fastsatt grenseverdi for stoffet)**

## 8. Ny grenseverdi

Dette kapitlet utarbeides etter at ASD har fastsatt den nye grenseverdien.



## 9. Referanse

Referanser som er vist til av SCOEL, og ellers i dette dokumentet, er listet under referanser i vedlagte kriteriedokument, side 36 (referanse 1, se vedlegg):

1. *Recommendation from the Scientific Committee on Occupational Exposure Limits for Bisphenol-A.* SCOEL/SUM/113, June 2014.



## 10. Vedlegg: SCOEL



# Recommendation from the Scientific Committee on Occupational Exposure Limits for Bisphenol-A

*SCOEL/SUM/113  
June 2014*

*Employment,  
Social Affairs  
and Inclusion*



## Table of Contents

1. Substance identification, physico-chemical properties.....	3
2. Occurrence/use and occupational exposure .....	4
3. Health significance .....	4
3.1. Toxicokinetics .....	4
3.1.1. Absorption.....	4
3.1.2. Metabolism .....	5
3.1.3. Excretion .....	5
3.1.4. Biological monitoring.....	6
3.2. Acute toxicity.....	8
3.3. Irritancy.....	8
3.4. Sensitisation .....	8
3.5. Endocrine modulating activity of BPA .....	9
3.6. Repeated dose toxicity .....	9
3.6.1. Human data.....	9
3.6.2. Animal data .....	9
3.7. Genotoxicity .....	11
3.7.1. Genotoxicity in vitro.....	11
3.7.2. Genotoxicity in vivo .....	11
3.8. Carcinogenicity .....	12
3.8.1. Human data.....	12
3.8.2. Animal data .....	12
3.9. Reproductive toxicity .....	13
3.9.1. Human data.....	13
3.9.2. Animal data .....	14
4. Recommendation .....	17
5. References.....	20



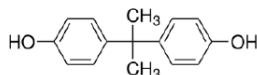
## Recommendation from the Scientific Committee on Occupational Exposure Limits for Bisphenol-A

8-hour TWA:	2 mg/m <sup>3</sup> (as inhalable dust)
STEL (15-min):	-
BLV:	-
BGV:	7 µg/l (urinary total bisphenol-A)
Additional categorisation:	-
Notation:	-

This Recommendation is based on the compilations by EFSA (2010 and 2014), EC (2003 and 2008) and WHO (2011). This was further supplemented by a literature search conducted by SCOEL in February 2014 covering the data published since the adoption of the previous evaluation of bisphenol-A by SCOEL in 2004.

### 1. Substance identification, physico-chemical properties

Chemical name: Bisphenol-A  
Synonyms (selected): 4,4'-Isopropylidenediphenol; 4,4'-dihydroxydiphenyl propane  
IUPAC name: 2,2-bis(4-hydroxyphenyl)propane  
Structural formula:



CAS No.: 80-05-7  
EC No.: 201-245-8  
Molecular formula: C<sub>15</sub>H<sub>16</sub>O<sub>2</sub>  
Molecular weight: 228.29  
Physical state at normal temperature and pressure: White solid flakes or powder (depends upon manufacturing process)  
Melting point: 155–157 °C (depends upon manufacturing process)  
Boiling point: 360 °C at 101.3 kPa (decomposition is also likely)  
Relative density at 25 °C: ca. 1.1–1.2 kg/m<sup>3</sup>  
Vapour pressure at 25 °C: 5.3 × 10<sup>-9</sup> kPa  
Solubility in water: 300 mg/l  
Partition coefficient: Log K<sub>ow</sub> ca. 3.3–3.5  
Flash point: ca. 207 °C  
Autoflammability: ca. 532 °C  
Explosive limits (in air): Minimum explosive concentration 0.012 g/l with O<sub>2</sub> > 5 %  
Oxidising properties: Not an oxidising agent

#### EU harmonised classification:

Skin sens. 1	H317	May cause an allergic skin reaction
Eye dam. 1	H318	Causes serious eye damage
STOT SE 3	H335	May cause respiratory irritation
Repr. 2	H361	Suspected of damaging fertility

The ECHA Risk Assessment Committee (RAC) recently proposed to strengthen the reproductive toxicity classification of bisphenol-A (BPA) to a category 1B (H360 May

damage fertility) reproductive toxicant regarding the adverse effects on sexual function and fertility.

## 2. Occurrence/use and occupational exposure

Four companies within the EU manufacture BPA. There are a total of six production sites based in Germany, the Netherlands, Belgium and Spain. The total amount of BPA manufactured within the EU was 1 438 kilotonnes in 2008. Global BPA consumption has increased at an average rate of almost 10 % per year from 2003 to 2006. However, since then the growth has slowed down and in Europe it is expected to be flat (Chemical Weekly 2009).

BPA is manufactured from phenol and acetone by an acid or alkaline catalysed condensation reaction. Its main use is in the production of polycarbonate plastics followed by use for manufacture of epoxy resins. These account for more than 95 % of the uses of BPA. Other uses include for example flame retardants, unsaturated polyester resins and polyacrylate, polyetherimide and polysulphone resins (EC 2008).

There are validated methods for the analysis of BPA from air samples (OSHA 2013, Bruhn 2012). Sampling is performed using sampling pumps and collection to glass fibre filters. After extraction (e.g. acetonitrile), samples are analysed using e.g. high or ultrahigh performance liquid chromatography.

BPA can be measured in urine or blood in the form of free, conjugated or total (free and conjugated) BPA. Usually, total BPA is measured from spot urinary samples. The most commonly used analytical methods include gas chromatography and liquid chromatography coupled with mass spectrometry (MS) or tandem MS (MS/MS). Also an ELISA assay is available for the detection of BPA from biological materials but the main disadvantage of this method is its reduced accuracy at low analyte concentrations due to cross-reactivity with other structurally related compounds (Fukata *et al* 2006).

## 3. Health significance

### 3.1. Toxicokinetics

#### 3.1.1. Absorption

The toxicokinetics of BPA after oral and parenteral administration has been well studied in rats and mice both *in vivo* and *in vitro*, and has been investigated to a lesser extent also in cynomolgus monkeys and humans (EC 2003, EFSA 2014, NTP 2008, WHO 2011). In the species studied, the available evidence shows that following oral administration, BPA is rapidly and extensively (about 85–100 % of the administered dose) absorbed from the gastrointestinal tract.

Dermal absorption has been studied *in vivo* or in *ex vivo* skin models. Morck *et al* (2010) reported 13 % absorption via the human skin. This study was performed according to OECD test guideline 428 but with an extended exposure period up to 48 hours. This result is rather well in accordance with a study by Demierre *et al* (2012), in which a penetration of 8.6 % with a maximum penetration rate of 0.022 µg/cm<sup>2</sup>/hour was measured in a test performed according to OECD guideline 428 and under GLP (good laboratory practise). Of the applied dose, 0.6 % was recovered from the remaining skin resulting in a total amount of bioavailable BPA of 9.3 %. This was calculated to mean that with an external exposure of 100 µg/day e.g. from thermal paper, an internal exposure of 9.3 µg may be reached (Demierre *et al* 2012). In a

third *ex-vivo* dermal penetration study (Kaddar *et al* 2008) percutaneous penetration of 4.1 % via pig skin was reported. In contrast to these studies, Zalko *et al* (2011) observed an absorption of 46 % via the human skin in a 72-hour culture system using human skin explants. The methodology applied differed, however, from the standard methodology for skin penetration testing.

Marquet *et al* (2011) measured an *in vivo* percutaneous absorption flux of 0.4  $\mu\text{g}/\text{cm}^2/\text{hour}$  in rats. According to their *ex vivo* studies on frozen human and rat skin, the permeability of human skin was 12-fold lower than that of rat skin. However, a 10-fold inter- and intraindividual variation was observed. Based on their results, it was calculated that a 1-hour occupational exposure over 2 000  $\text{cm}^2$  may lead to absorption of 4  $\mu\text{g}/\text{kg}/\text{day}$ .

There were no data on the toxicokinetics of BPA following inhalation exposure, but it is assumed that appreciable absorption would occur.

### 3.1.2. Metabolism

After oral dosing, BPA is removed rapidly from the blood by first pass metabolism in the liver. In controlled oral dosing studies in humans using labelled BPA, free (unconjugated) BPA represented only 0.2–1.2 % of the total AUC (area under the curve) of BPA in blood or < 2 % of the total maximum concentration ( $C_{\text{max}}$ ) (Taylor *et al* 2011, Volkel *et al* 2005 and 2011). However, the route of exposure is of paramount importance as there are marked differences in free BPA concentrations after oral as compared to parenteral administration of an equivalent dose (EC 2003). The bioavailability of free BPA can be 6–240-fold higher after intraperitoneal or subcutaneous dosing than after oral dosing (Pottenger *et al* 1997a,b). These differences may explain some effects seen after parenteral dosing but not after oral dosing.

Doerge *et al* (2010a,b, 2011b) have studied comparative kinetics of BPA in adult mice, rats and monkeys. According to their data, the levels of free (unconjugated) BPA in serum after single oral 100- $\mu\text{g}/\text{kg}$  bw exposures are very low in all species: the AUC being 0.1 nMh in mice (0.2 % of the dose), 2.6 nMh in rats (2.8 %) and 1.5 nMh (0.9 %) in monkeys. This is well in accordance with the human data presented above. No comparative data on the levels of free BPA after inhalation exposure were available. It should be noted that a significant portion of inhaled BPA aerosols may actually become ingested.

There are contradictory data on the ability of the viable skin to metabolise BPA (Zalko *et al* 2011, Marquet *et al* 2011). According to *in vitro* studies, first-pass metabolism of BPA does not occur in lungs (Mazur *et al* 2010, Trdan Lusin *et al* 2012).

The major metabolic pathway in all species studied involves conjugation of BPA with glucuronic acid. In addition to the glucuronidation pathway, *in vivo* and *in vitro* studies suggest that BPA may be subject to limited oxidation to bisphenol *O*-quinone by cytochrome P450, and also to conjugation with sulphate.

### 3.1.3. Excretion

The major route of excretion in the rat and mouse is via faeces. The available data indicate that the percentage of the administered dose recovered in the faeces is in the range of 50–83 %. Urinary excretion is of secondary importance in the rat, with 13–42 % of the administered dose being recovered in the urine. Over 7 days post-dosing, 70–80 % of the administered dose was excreted in the faeces in rats. Elimination was rapid; the majority of the dose was excreted by 72 hours post-dosing. A sex difference

was also observed in rats in urinary excretion, with females excreting approximately twice as much radioactivity (24–28 %) as males (14–16 %). In addition, a strain difference was observed, with female F344 rats excreting approximately twice as much radioactivity in the urine as female CD rats (EC 2008). Data from a number of studies suggest limited excretion of BPA in the milk. Doerge *et al* (2010c) evaluated the lactational transfer of BPA after repeated oral dosing in rats and noted that even when free BPA was detected in all dam serum and milk samples, levels in pup serum were below the detection limits, and calculated doses delivered to pups lactationally were 300-fold lower than the dose administered to the dams. In rats and monkeys, free BPA has been shown to cross the placenta following oral administration, but the foetal levels have been in the same range or lower than those in maternal tissues (Doerge *et al* 2011a, Patterson *et al* 2013). *Ex vivo* studies using human placentas suggest a transplacental transfer rate of 1 (Mose *et al* 2012, Balakrishnan *et al* 2010).

In contrast to the findings in rodents, 84–97 % of a BPA dose administered to humans is excreted as glucuronide or sulphate conjugates in urine within a few hours (5–7 hours) after the administration. Within 24 hours, recovery from the urine is increased up to 100 % (Volkel *et al* 2002 and 2005). Free urinary BPA is only rarely detected in the general population (Volkel *et al* 2008). These interspecies differences in the main route of excretion of BPA have been explained by the differences in the thresholds for biliary elimination; the molecular weight of BPA-glucuronide is above the threshold in rats (approximately 350 Daltons) but below the threshold in humans (about 550 Daltons). Enterohepatic circulation in rodents accounts for the longer elimination half-life in rodents as compared to humans.

#### 3.1.4. Biological monitoring

BPA has been a subject for several biomonitoring studies among the general population.

Results of several population studies measuring BPA levels in the general population were recently reviewed by e.g. Vandenberg *et al* 2010. Measurable levels of total BPA is usually present in the urine of most subjects among the general population. The levels measured in general, mostly adult, populations in USA, Canada, Germany and Finland are presented in Table 1.

The German Federal Environment Agency recently set a reference value of 7 µg/l for 20–29-year old adults (UBA 2012). This is based on the 95<sup>th</sup> percentile of total urinary BPA in a reference population of 600 20–29-year old adults. Children usually have higher BPA levels than adolescents who in turn have higher levels than adults (Calafat *et al* 2008).

Limited data is available on the BPA biomarker levels in occupationally exposed populations. Hanaoka *et al* (2002) reported urinary BPA (total BPA) levels in epoxy resin sprayers. Median levels were 1.06 (range ND–11.2 µmol/mol creatinine, n = 42), whereas the median in the control group in this study was 0.52 (range ND–11.0 µmol/mol creatinine, n = 42). Regardless of the statistically significant difference between the sprayers and the control group, the range of measured values was similar in both groups. He *et al* (2009) studied BPA exposure in Chinese workers in epoxy resin and BPA manufacturing facilities by air monitoring and by measuring urinary BPA levels. BPA was detected in 96 % of the air samples and the median concentration was 6.67 µg/m<sup>3</sup>. Measurable levels were detected both at epoxy resin manufacturing and BPA manufacturing (median 7.89 and 4.72 µg/m<sup>3</sup>, respectively). Pre-shift and post-shift urinary samples were collected. In resin manufacturing, median pre- and post-shift levels were 80.2 and 108 µg/g creatinine, respectively (n = 178 and 191),



**Table 1.** Total urinary BPA levels (after hydrolysis) in the general population.

Study (country)	Study population and sample size	BPA level (µg/l)	
		GM	95th percentile
Calafat <i>et al</i> 2005 (NHANES 1988–94, USA)	184 males, 210 females	1.33	5.18
Calafat <i>et al</i> 2008 (NHANES 2003–04, USA)	9– < 60 year-old males and females (n = 2 517)	2.6	15.9
	20–59 year-old males and females (n = 951)	2.6	15.9
CDC 2012 (NHANES 2005–06, USA)	> 20 years (n = 1 490)	1.75	10.7
CDC 2012 (NHANES 2007–08, USA)	> 20 years (n = 1 814)	1.99	13.3
CDC 2012 (NHANES 2009–10, USA)	> 20 years (n = 1 914)	1.79	9.60
Koch <i>et al</i> 2012 (Germany)	20–29 years (n = 600)	1.55	7.37
Health Canada 2010 (Canada)	20–39 years (n = 1 165)	1.33	7.30
	40–59 years (n = 1 219)	1.04	6.58
Porrás <i>et al</i> 2014 (Finland)	22–67 years (n = 121)	2.6	8.1

and in BPA manufacturing 170 and 233 µg/g creatinine (n = 8 and 7). Correlation analysis of 131 workers who contributed urine samples both pre- and post-shift showed that there was a significant correlation between levels of personal airborne BPA and urinary BPA pre-/post-shift levels. The main pollution sources were said to be crushing, feeding and packing workstations. Although not discussed in the report, skin (including skin to mount) exposure may have contributed to urinary levels. In addition, there were some discrepancies in the reported air and urinary levels in the report. In non-occupationally exposed Chinese males (n = 419), median urinary BPA levels of 1.43 µg/g creatinine were reported by the same research group. The 75<sup>th</sup> percentile was 14.18 µg/g creatinine (He *et al* 2009).

Wang *et al* (2012b) reported average urinary BPA concentrations of 55.73 ± 5.48 ng/ml (range 5.56–1 934.85 ng/ml) among 28 Chinese workers exposed to BPA in epoxy resin manufacturing. Brill (2013) measured levels up to 2 062 µg/l among workers in a BPA processing plant in Germany. In a Finnish study, levels up to 1 500 µg/l were seen in thermal paper manufacturing (Porrás *et al* 2014).

Krishnan *et al* (2010) estimated the concentration of BPA in urine corresponding to the tolerable daily intake set by EFSA in 2006 (0.05 mg/kg) on the basis of available data on BPA toxicokinetics after oral exposure. This is called a biomonitoring equivalent. Taking into account that BPA is almost completely eliminated from the blood into urine after oral exposure, a biomonitoring equivalent of 2.0 mg/l (2.6 mg/g creatinine) was calculated using the following formula:  $C_v = D \times BW \times F_{UE} / V$ , where  $C_v$  is the average urinary BPA concentration on a volume basis, D is a unit dose of BPA at the tolerable daily intake (TDI) level, BW is the body weight for the group,  $F_{UE}$  is the urinary excretion fraction (= 1 for BPA), i.e. the fraction of the applied dose excreted in the urine, and V is the 24-hour average urinary volume.

In Germany, a biological limit value (BLV) for occupational exposure of 80 mg/l has been set for total BPA in urine (DFG 2013).

### 3.2. Acute toxicity

No useful information was available on the acute toxicity of BPA in humans. Oral LD<sub>50</sub> values beyond 2 000 mg/kg are indicated in the rat and mouse, and dermal LD<sub>50</sub> values above 2 000 mg/kg are evident in the rabbit (Hazleton Laboratories 1985, Mellon Institute 1948 and 1965, NTP 1982). For inhalation, a 6-hour exposure to 170 mg/m<sup>3</sup> (the highest attainable concentration) produced no deaths in rats; slight and transient nasal tract epithelial damage was observed (Nitschke *et al* 1985b). These data indicate that BPA is of low acute toxicity by all routes of exposure relevant to human health.

### 3.3. Irritancy

Limited human anecdotal information of uncertain reliability is available from written industry correspondence suggesting that workers handling BPA in the past experienced skin, eye and respiratory tract irritation (Dow Chemical 1957, Du Pont 1962). It cannot be determined whether the reported skin reactions were related to skin sensitisation (see Section 3.4) or irritation. However, a well conducted animal study clearly showed that BPA is not a skin irritant (Leuschner 2000b). The same research group also showed in a well conducted animal study that BPA is an eye irritant; effects persisted until the end of the study (day 28 post-instillation) in 1 of 3 rabbits (Leuschner 2000a). Overall, taking into account the animal and human evidence, BPA has the potential to cause serious damage to the eyes.

Slight and transient nasal tract epithelial damage was observed in rats exposed to BPA dust at 170 mg/m<sup>3</sup> (the highest attainable concentration) for 6 hours (Nitschke *et al* 1985b). These data suggest that BPA appears to have a limited respiratory irritation potential.

### 3.4. Sensitisation

With respect to skin sensitisation in humans, there are several reports of patients with dermatitis responding to BPA in patch tests (EC 2003). However, it is unclear whether BPA or related epoxy resins were the underlying cause of the hypersensitive state. Anecdotal information indicates skin inflammation in workers handling BPA although, given the uncertain reliability of this information, no conclusions can be drawn from it. In animals, a skin sensitisation test performed according to current regulatory standards is not available. The available studies are negative, but the test reports lack detail and no reliable justifications were given for the choice of concentrations used (Thorgeirsson and Fregert 1977, Procter and Gamble Co. 1969). It is possible that the concentrations used in all the available studies were not maximised and a greater response might have been obtained with higher induction and challenge concentrations. Based on the findings from the most robust study, BPA may possess a skin sensitisation potential, albeit a limited one. BPA in the presence of UV light can also elicit skin responses in humans, and reproducible positive results for photosensitisation have been obtained in the mouse ear swelling test (Allen and Kaidbey 1979, Gerberick and Ryan 1990, Maguire 1988). Therefore, examination of the available human and experimental animal studies leaves the picture somewhat unclear as to whether one or more of the following are properties of BPA; (1) orthodox skin sensitisation (2) photosensitisation (3) BPA eliciting a response in people previously skin sensitised to another substance (e.g. epoxy resins). Thus, the precise nature of the hazardous properties of BPA on the skin is unclear, but clearly skin reactions can be a potential consequence of repeated skin exposure in humans. Overall, taking all of the available data into account, BPA is considered capable of

producing skin sensitisation responses in humans. There are no data from which to evaluate the potential of BPA to be a respiratory sensitiser.

### 3.5. Endocrine modulating activity of BPA

BPA has been shown to have endocrine modulating activity in a number of *in vitro* and *in vivo* screening assays (EC 2003). The potency of this activity in these assays generally ranged from 3–5 orders of magnitude less than that of oestradiol. Regardless of the low activity compared to oestradiol, these endocrine-modulating effects of BPA have been the main concern and subject for BPA related research during the past few years. These effects have been suggested to occur at very low dose levels and exhibit so-called non-monotonic dose-response curves (NMDRCs), defined as a nonlinear relationship between dose and response where the slope of the curve changes sign somewhere within the range of doses examined (Vandenberg *et al* 2012). The scientific validity of the non-monotonic dose-response relationship has been questioned. However, non-monotonic dose-response has been observed at un-physiological conditions, which are not relevant for risk assessment. Toxic effects suggested to be related to the endocrine modulating activity of BPA are discussed below under the respective chapters: repeated dose toxicity (metabolic effects), carcinogenicity (hormonal cancers) and reproductive toxicity.

### 3.6. Repeated dose toxicity

#### 3.6.1. Human data

There are some recent cross-sectional studies on the general population reporting associations between urinary BPA levels and diabetes, obesity or cardiovascular diseases (Carwile and Michels 2011, Eng *et al* 2013, Melzer *et al* 2012a,b, Shankar *et al* 2011, Shankar *et al* 2012a,b,c, Silver *et al* 2011, Trasande *et al* 2012, Wang *et al* 2012a, all reviewed in EFSA 2014). Because of the cross-sectional nature of these studies and possible confounding by diet (which is also the main source of BPA) or other concurrent exposures, no conclusions on the association can be made based on these studies. In addition, an association was not supported by the re-analysis of the NHANES (National Health and Nutrition Examination Survey) based data sets with control of all relevant confounders (Lakind *et al* 2012). The authors of this study emphasised the inappropriateness of using cross-sectional datasets like NHANES to draw conclusions about the causal associations between short-lived environmental chemicals like BPA and chronic complex diseases.

Wang *et al* (2012b) reported a cross-sectional analysis of the relationship between urinary BPA concentrations and blood or urinary markers of liver function, glucose homeostasis, thyroid function and cardiovascular diseases among 28 Chinese workers exposed to BPA in epoxy resin manufacturing. The average urinary BPA concentration was  $55.73 \pm 5.48$  ng/ml (range 5.56–1934.85 ng/ml). Higher urinary BPA concentrations were associated with a significant increase in FT3 (free triiodothyronine) levels in this group of workers. A possible effect of other, confounding exposures cannot be ruled out. No conclusions can be made on the basis of this single, small study.

#### 3.6.2. Animal data

In animals, there were no data relating to repeated dermal exposure. Repeat inhalation studies were available in the rat (Nitschke *et al* 1985a and 1988). The principal effect was the same as that observed following a single exposure - slight upper respiratory tract epithelium inflammation. Very slight to slight inflammation and

hyperplasia of the olfactory epithelium were observed following exposure to 50 and 150 mg/m<sup>3</sup> [6 hours/day, 5 days/week for 2 or 13 weeks; 150 mg/m<sup>3</sup> is close to the highest attainable concentration; the particle mass median aerodynamic diameter (MMAD) was 2–6 µm], and a NOAEC of 10 mg/m<sup>3</sup> was identified in rats in this 13-week study.

In early 90-days studies in rats, a decrease in body weight gain and minor changes in organ weights at 100 mg/kg/day and above were seen after dietary administration (NTP 1982, Til *et al* 1978). Dietary studies in mice indicated that the liver is a target organ in this species with changes being observed in the size and nucleation state of hepatocytes in 2-year and 90-day studies (Furukawa *et al* 1994, NTP 1982). The incidence and severity of these treatment-related multinuclear giant hepatocytes were markedly greater in males than in females. It was not possible to identify a no-effect level for males, the effect being observed at all dose levels used from the lowest dose tested of 120 mg/kg/day (2-year study). Even at this lowest dose level, a large proportion (84 %) of the animals examined showed signs of this effect. In females, a no-effect level of 650 mg/kg/day was identified for these cellular changes in the 2-year study.

The studies providing relevant dose-response data on repeated dose toxicity after oral exposure include also the multigeneration and 2-generation studies by Tyl *et al* (2002 and 2008) in rats and mice. Tyl *et al* (2002) studied the effects of dietary levels of 0, 0.015, 0.3, 4.5, 75, 750 and 7 500 mg/kg feed (corresponding to an intake of 0.001, 0.02, 0.3, 5, 50 and 500 mg/kg bw/day of BPA) in Sprague-Dawley rats over three offspring generations. Adult systemic toxicity was evident at the two highest doses of 50 and 500 mg/kg bw/day in all generations. The effects included reductions in body weights and weight gains, which were evident in males already at 50 mg/kg/day. At necropsy, F0, F1, and F2 parental and F3 retained adult absolute non-reproductive organ weights were almost uniformly reduced for liver, kidneys, adrenal glands, spleen, pituitary and brain at 500 mg/kg bw. Slight to mild renal tubular degeneration and chronic hepatic inflammation were observed at a higher incidence in F0, F1 and F2 females at 500 mg/kg bw. No effects on food consumption were seen and no treatment or dose-related effects were seen in clinical observations. There were no toxicologically significant effects on these parameters at 5 mg/kg bw/day (NOAEL).

In a 2-generation study in mice (Tyl *et al* 2008) at dietary doses of 0, 0.018, 0.18, 1.8, 30, 300 or 3 500 mg/kg feed (corresponding to an intake of 0, 0.003, 0.03, 0.3, 5, 50 or 600 mg BPA/kg bw/day), effects on liver were observed in F0/F1 adult males in the two highest dose groups. The effects included increased weights of the liver at 600 mg/kg bw/day and increased incidence of liver centrilobular hepatocyte hypertrophy at 50 mg/kg bw (minimal severity) and at 600 mg/kg bw (minimal to mild severity). Also renal nephropathy with minimal severity was seen at the highest dose. Absolute kidney weights were increased in both F0 and F1 males at the two highest dose levels and a small (statistically significant) increase was seen also at the dose levels of 0.3 and 5 mg/kg in F1 males only (not in F0 or in retained F1 males). Reduced body weights were seen in males at the highest dose without any effects on food consumption. In females, increased absolute and/or relative weight of the liver and kidneys and centrilobular hepatocyte hypertrophy of minimal severity were seen at the highest dose level. The authors suggested a NOAEL of 5 mg/kg bw based on liver effects. Effects on kidney weights in male mice at the dose levels of 0.3–50 mg/kg bw were not considered relevant by the authors because of the lack of accompanied histological changes, lack of clear dose-response relationships or lack of clear effects in F0 or retained F1 male mice. EFSA (2014) calculated the benchmark dose level corresponding to 10 % extra risk for liver and kidney effects (BMDL<sub>10</sub>) based on this study. BMDL<sub>10</sub> for hepatocyte hypertrophy was 3 460 µg/kg bw and for increased kidney weights 3 633 µg/kg bw (right kidney) and 3 887 µg/kg bw (left



kidney). Kidney weight changes were considered more relevant than possibly adaptive hepatocyte hypertrophy. It should be noted, however, that in rats (Tyl *et al* 2002), a reduction rather than an increase in kidney weights were seen. Altogether, these data show that liver and kidneys are the main target organs of BPA after high oral daily doses > 5 mg/kg bw.

In dogs, a 90-day dietary study showed a no-effect level of approximately 80 mg/kg bw/day, with increases in relative liver weight observed at approximately 270 mg/kg bw/day (General Electric 1976).

In addition, there are several animal studies using lower doses of BPA (< 5 mg/kg bw) claiming that BPA has an effect on lipogenesis (causing obesity), and glucose or insulin regulation (resulting in diabetes). These studies were reviewed by EFSA (2014). Many of these employed prenatal exposure, but some studies were performed also in animals exposed postnatally. Results from these studies are, however, inconsistent, and even if some changes in glucose and insulin regulation or pancreas have been seen in short-term studies, these are not supported by the findings related to diabetes or obesity in long-term term animal studies (EFSA 2014).

### 3.7. Genotoxicity

#### 3.7.1. Genotoxicity *in vitro*

BPA appears to have demonstrated aneugenic potential *in vitro*, positive results being observed without metabolic activation in a micronucleus test in Chinese hamster V79 cells and in a non-conventional aneuploidy assay in cultured Syrian hamster embryo cells (Pfeiffer *et al* 1997, Tsutsui *et al* 1998). Additionally, in cell-free and cellular systems, there is information showing that BPA disrupts microtubule formation and spindle apparatus, which may result in aneuploidy (EC 2003, NTP 2008). However, these effects have not been unequivocally demonstrated *in vivo* (NTP 2008). BPA produced adduct spots in one post-labelling assay with isolated DNA and a peroxidase activation system, but it does not appear to produce either gene mutations or structural chromosome aberrations in bacteria, fungi or mammalian cells *in vitro* (EC 2003, NTP 2008).

#### 3.7.2. Genotoxicity *in vivo*

No human data regarding genotoxicity were available. The standard mouse bone marrow micronucleus test gave negative result (Shell Oil Company 1999). Pacchiorotti *et al* (2008) found no increase in chromosomal aberrations in germ cells or in bone marrow cells of rats after acute, sub-chronic or chronic *in vivo* exposure. Female mice were orally treated with BPA, either a single dose with 7 daily administrations or for 7 weeks to BPA in drinking water. No significant induction of hyperploidy or polyploidy was observed in oocytes and zygotes at any treatment condition. With male mice, no delay of meiotic divisions was found in the BrdU assay after 6 daily oral doses of BPA and no induction of hyperploidy and polyploidy in epididymal sperm was seen after 6 daily oral BPA doses. Finally, 2 daily oral BPA doses did not induce any increase in micronucleus frequencies in polychromatic erythrocytes of mouse bone marrow. The doses used were up to 20 mg/kg bw in the single dose study, and 0.002–0.2 mg/kg bw per day in the repeated dose studies. Doses were selected on the basis of the Hunt *et al* (2003) study showing aneuploidy in mice oocytes *in vivo*.

Lack of clastogenic effects were supported also by e.g. Naik *et al* (2009) who did not see any increases in chromosomal aberrations or micronuclei in mice bone marrow cells after oral exposure. However, increased incidences of c-mitosis were seen at dose levels of 50 and 100 mg/kg suggesting an effect on the mitotic spindle

apparatus. This effect is supported by *in vitro* studies by Johnson and Parry (2008) and Tayama *et al* (2008) showing increased incidences in binucleate-micronucleate cells and c-mitosis *in vitro*, respectively, at dose levels not causing cytotoxicity. An effect on microtubules can be considered, however, to have a dose threshold. Considering all of the available genotoxicity data and the absence of significant tumour findings in animal carcinogenicity studies (see Section 3.8), it does not appear that BPA has significant mutagenic or genotoxic potential *in vivo*.

### 3.8. Carcinogenicity

#### 3.8.1. Human data

There were no human data contributing to the assessment of whether or not BPA is carcinogenic.

#### 3.8.2. Animal data

In animals, a dietary carcinogenicity study in two species, F344 rats and B6C3F<sub>1</sub> mice, was available (NTP 1982). A small increased incidence of leukaemias was seen in male and female F344 rats along with increases in the frequency of mammary gland fibroadenomas in male rats. These increases were not statistically significant, were slight and in a strain prone to these tumours. An increased incidence in benign Leydig cell tumours seen in male rats was within historical control limits. In mice, a small increased incidence in lymphomas was observed in males, but was not statistically significant and there was no dose-related trend. No increased incidence in any tumour type was observed in female mice. Overall, all of these tumour findings in rats and mice were considered not toxicologically significant. Consequently, it was concluded that BPA was not carcinogenic in this study in either species.

No inhalation or dermal carcinogenicity studies were available, although in repeat exposure inhalation toxicity studies, BPA did not exhibit properties that raise concern for potential carcinogenicity. Only minimal inflammation was seen in the upper respiratory tract at 50 mg/m<sup>3</sup> in a 13-week study, and the severity did not increase up to concentrations close to 150 mg/m<sup>3</sup>, the maximum attainable concentration in the experimental system used. Taking into account all of the animal data available, the evidence suggests that BPA does not have carcinogenic potential.

Recently, concerns have been raised for the possible contribution of BPA on prostate and mammary gland development rendering these organs more susceptible to neoplasia especially when exposed during neonatal age (Acevedo *et al* 2013, Betancourt *et al* 2010, Jenkins *et al* 2009 and 2011, Lamartiniere *et al* 2011, Moral *et al* 2008, Prins *et al* 2011, Tharp *et al* 2012, Timms *et al* 2005, Vandenberg *et al* 2013, Weber Lozada and Keri 2011). These studies were performed in the "low dose range" i.e. at levels well below 5 mg/kg bw. Most of these studies did not study the effect on tumour incidence but rather on tissue/cell proliferation. Some of these suggest normal monotonic dose-response relationships for these carcinogenic or proliferative effects while in other studies a non-monotonic dose-response curve is claimed. The studies evaluating the effects on cancer incidence include the study by Jenkins *et al* (2009) in which rats were exposed lactationally to low doses (25 and 250 µg/kg) of BPA and a single oral dose of the tumour inducer dimethylbenzanthracene (DMBA). A dose-related increase in the number of tumours was seen, which was statistically significant at the highest dose. A decrease in tumour latency was also reported. Similar results were reported also by Betancourt *et al* (2010) with both prenatal and postnatal exposure to BPA. However, in a subsequent study by Jenkins *et al* (2011), in which adult transgenic mice were exposed to 0.5, 5, 50 or 500 µg/kg of BPA from week 8 to week 36 of age, an increase in the number of tumours per mouse and in the number

of mice with metastases were observed only at the 1–2 lowest dose levels without any clear dose-response relationship. No data on the exact number of mice with tumours were given. Acevedo *et al* (2013) studied the effect of subcutaneous prenatal or both pre- and postnatal exposure to BPA on mammary gland cell proliferation and neoplasia at the dose levels of 0.25, 2.5 or 250 µg/kg. Some preneoplastic lesions and single cancers (0–1 cancers per 23–33 animals) were detected in the treated groups without any significant increase or dose-response relationship. The highest dose level resulted in measurable serum levels of free BPA in pups exposed via lactation. Lower dose levels were estimated to result in such low serum levels that they would not have been detected with current analytical methods (and would not have been distinguishable from background levels measured in untreated animals). All these studies, however, suffer from deficiencies in design or execution, which have been discussed in detail in EFSA 2014. The US Food and Drug Administration/National Center for Toxicological Research (FDA/NCTR) is currently conducting a long-term study in rats in order to evaluate the potential carcinogenicity of BPA after pre- and postnatal exposure. Related to this long-term study, a shorter duration reproductive study was performed, in which rats were exposed from gestation day 6 to postnatal day 90 to a wide-range of oral doses (0.0025–300 mg/kg) of BPA (Delclos *et al* 2014). At postnatal days 21 and 90, an increase in mammary gland duct hyperplasia was seen in the three highest dose groups (2.7, 100 and 300 mg/kg) and in females only. Variability in incidence across the dose range was seen. The severity of hyperplasia was minimal. It should be noted that pups were dosed directly by gavage from postnatal day 1 instead of being exposed via dam's milk. Thus, the postnatal exposure was higher than in many other studies involving postnatal exposure via dam's milk. Ethinyl oestradiol at the doses of 0.5 and 5 µg/kg, in contrast, induced mammary gland duct hyperplasia especially in males, which is in accordance with earlier studies on the effects of oestrogenic agents.

Thus, there is currently no convincing evidence of carcinogenicity of BPA when administered either during the adulthood or perinatally. However, as concluded by EFSA (2014), there are some data (including the data by Delclos *et al* 2014) that raise some concern for BPA effects on mammary gland cell proliferation after pre- and perinatal exposure. Whether this is linked to increased cancer incidence in later life or not remains to be shown.

### 3.9. Reproductive toxicity

#### 3.9.1. Human data

In a study conducted in China, Li *et al* (2010b) examined the effect of occupational BPA exposure on male reproductive function. Workers (n = 164) were exposed to mean air levels of 0.006 mg/m<sup>3</sup> of BPA, the highest levels being in packaging operations (geometric mean 0.016 mg/m<sup>3</sup>), and their sexual function was evaluated using a standardised male sexual function inventory. BPA exposed workers reported higher levels of reduced sexual desire (OR 3.9), erectile or ejaculation difficulty (ORs 4.5 and 7.1, respectively), and reduced satisfaction with their sex life (OR 3.9). A dose-response relationship with cumulative BPA exposure was seen. When sexual function among these workers was correlated with urinary BPA levels (based on two spot samples, before and after the work shift), a significant correlation between urinary BPA levels and self-reported sexual dysfunction was seen. The median urinary BPA level was 53.7 µg/g creatinine (with an interquartile range of 8.6–558.9 µg/g creatinine) among the exposed workers (Li *et al* 2010a). In their third study, Li *et al* (2011) reported a statistically significant association between increasing urinary BPA levels and decreasing sperm concentration, total sperm count, sperm vitality and motility among 218 men working in the same factories. Compared to those men who

had no detectable urinary BPA, those with detectable urinary BPA had an OR of 3.4 for lower sperm concentration, an OR of 3.3 for lower sperm vitality, an OR of 4.1 for lower sperm count and an OR of 2.3 for lower sperm motility. Among the highest tertile of BPA exposure, higher ORs for these effects were detected. An inverse correlation between urinary BPA levels and sperm concentration and sperm count was noted also among environmentally exposed persons ( $n = 88$ ). Although some confounders had been taken into account in these occupational studies, it is not possible to exclude an effect of other occupational exposures on the studied parameters. Also a selection bias cannot be excluded since only 58 % of the invited men participated in the study. The latest cross-sectional study from China among petrochemical industry workers ( $n = 137$ ) reported decreased androstenedione and free testosterone levels, decreased free androgen index and increased sex-hormone binding globulin levels when compared to the control group. Inhibin, follicle-stimulating hormone (FSH), prolactin, oestradiol and total testosterone levels were unchanged. The exposed group had a median serum BPA level of 3.198 compared to 0.276  $\mu\text{g/l}$  in controls (Zhou *et al* 2013).

Cha *et al* (2008), on the other hand, reported decreased testosterone levels and increased luteinising hormone (LH) and FSH levels among 25 epoxy resin painters with increased urinary BPA levels (2.61  $\mu\text{g/g}$  creatinine vs. 1.38  $\mu\text{g/g}$  creatinine in controls). This contrasts with the findings of Hanaoka *et al* 2002 who showed decreased FSH levels among 42 epoxy resin sprayers with slightly elevated urinary BPA levels. It should be noted that the difference in BPA levels between exposed and non-exposed was very small.

Regarding developmental effects, Miao *et al* (2011a,b) reported decreased birth weight and shortened anogenital distance (in boys) in the offspring of occupationally BPA exposed mothers. Characterisation of the exposure was limited and potential confounding by diet or other exposures cannot be excluded. There are also other, mainly cross-sectional studies which have suggested a linkage between maternal BPA exposure from environmental sources and pregnancy outcome (foetal growth etc.). However, the effect of diet or other concurrent exposures on pregnancy outcome cannot be ruled out. Thus, no firm conclusions on causal association can be drawn from these studies (EFSA 2014).

Braun *et al* (2009 and 2011) examined the relationship between gestational BPA exposure (measured as serial urinary BPA samples) and neurobehavioral effects in infants in a prospective study. An association between BPA levels and externalising behaviours (aggression, hyperactivity) among 2-year old girls was noted. At the age of 3, the girls showed a more anxious and depressed behaviour and poorer emotional control and inhibition. However, another prospective study observed an association with aggressive behaviour and emotional reactivity only in boys (Perera *et al* 2012), whereas in some other studies no association with child behaviour was seen (Miodovnik *et al* 2011, Yolton *et al* 2011). Thus, no conclusions on the possible association between pre- or early postnatal exposure to BPA and behaviour can be made based on the human data.

### 3.9.2. Animal data

The effects of BPA on fertility and reproductive performance have been investigated in three good quality 3-generation and 2-generation studies in rats or mice, and one older continuous breeding study in mice. Since there is an ongoing discussion on possible low-dose effects and non-monotonic dose-response curves, three of these studies employed also low dose ranges.



The oldest one of these reproductive toxicity studies is a continuous breeding study in mice, which provides some evidence that BPA can cause adverse effects on fertility at high dose levels (NTP 1985a). In the F0 generation, no effects on fertility were seen at 300 mg/kg bw/day, but at dose levels of approximately 600 mg/kg bw/day and above, reductions in the numbers of litters produced, litter size and numbers of live pups per litter were observed in each of the 4–5 litters produced. These effects were observed in the absence of significant parental toxicity. In contrast, no adverse effects on fertility were observed in the single litter tested at each dose level from the F1 generation. A small but statistically significant and dose related decrease in epididymal weight was seen at all doses in the F1 generation, but the significance of this finding is uncertain because a comparable effect was not seen in F0 mice. In spite of the uncertainty, the epididymis is associated with sperm transport and storage, and any reduction in the weight of this organ would be of concern.

In the 3-generation study, an effect on fertility (reduction in litter size) was seen in all three generations at the top dose of 500 mg/kg bw (Tyl *et al* 2002). Although this effect was seen only at a dose level causing parental toxicity (a reduction in body weight gain of > 13 %) in both sexes and renal tubule degeneration in females, it is not clear whether or not the finding could be a secondary consequence of parental toxicity, or a direct effect of BPA. Reductions in body weights and weight gains were seen in males already at 50 mg/kg bw. No effects on fertility were seen at 50 mg/kg or at lower dosages (0.001–5 mg/kg bw/day). Regarding developmental effects, a statistically significant decrease in mean pup body weight gain with concomitant delays in the acquisition of developmental landmarks (vaginal patency and preputial separation) was observed at 500 mg/kg bw on postnatal days 7–21 in males and females of all generations (F1–F3) (Tyl *et al* 2002). These decreases in pup body weight gain and delays in development were seen in the presence of maternal toxicity. No treatment-related effects were reported in the offspring of animals exposed to 50 mg/kg bw. The NOAEL for reproductive and developmental endpoints was 50 mg/kg bw/day and for systemic toxicity 5 mg/kg bw/day.

In a 2-generation study in mice, no effects on adult mating, fertility or gestational indices, ovarian primordial follicle counts, oestrous cyclicity, precoital interval, sperm parameters or reproductive organ weights or histopathology (including the testes and prostate) were seen in the dose range of 0.003–600 mg/kg bw (Tyl *et al* 2008). Signs of systemic toxicity (e.g. liver centrilobular hepatocyte hypertrophy) were seen at the doses of 50 mg/kg bw and higher (see Section 3.6). However, reduced F1/F2 weanling body weight, reduced weanling spleen and testes weights (with seminiferous tubule hypoplasia), slightly delayed preputial separation (PPS), and an increased incidence of undescended testes in weanlings were seen at the highest dose level of 600 mg/kg bw/day. The latter finding was considered as a developmental delay in the normal process of testes descent since it did not result in impaired reproductive performance later in life (Tyl *et al* 2008). Offspring sex ratios or postnatal survival was unaffected. The NOAEL for developmental effects was 50 mg/kg bw/day and for systemic toxicity 5 mg/kg bw/day. No effects were seen in the low dose range of 0.003–5 mg/kg bw/day.

Also in a 2-generation rat study employing low doses of 0.2–200 µg/kg bw/day by gavage with endocrine-sensitive and neurobehavioural end-points, no effects on any reproductive or developmental parameters were seen at any dose level (Ema *et al* 2001).

In a repeated dose study by FDA/NCTR (Delclos *et al* 2014), rats were exposed from gestation day 6 to postnatal day 90 to a wide-range of oral doses (0.0025–300 mg/kg bw) of BPA. Body weights of high-dose males and females, and of the second highest-dose (100 mg/kg) females were decreased when compared to the controls. No effect

on male sperm production, reproductive organ weights or histopathology was seen on postnatal day 90. A 1–2 days delay in testicular descent was seen in the 0.26- and 300-mg/kg dose groups but not in others (including 2.6 or 100 mg/kg). In females, higher rates of abnormal cyclicity, decreased ovarian weights and depletion of corpora lutea and antral follicles were detected at the highest dose level (300 mg/kg). Increased serum oestradiol and prolactin and decreased progesterone levels were seen in females at postnatal day 80. No treatment related effects were seen in markers of sexual developmental in either sex. Ethinyl oestradiol at the dose levels of 0.5 and 5 µg/kg induced several changes in the above mentioned parameters. BPA did not affect obesity or glucose metabolism parameters (fat pad weights, serum glucose, insulin) either.

Regarding standard developmental endpoints, no effects were seen in old standard developmental studies in rats and mice. In rats, a maternal LOAEL and a foetal NOAEL of 160 and 640 mg/kg bw/day, respectively, were identified (Morrisey *et al* 1987, NTP 1985c). In mice, maternal and foetal NOAELs were 250 and 1 000 mg/kg bw/day, respectively (NTP 1985b).

In contrast to these guideline based studies described above, there are several other studies on the effects of low BPA doses (< 5 mg/kg bw/day) on the reproductive parameters. Some of these claim a non-monotonic dose-response curve, whereas in others, a monotonic dose-response relationship has been seen.

In early studies in mice, adverse effects on male reproductive tract development (an increase in prostate weight in two studies and a reduction in epididymis weight in one study) were reported after gestational exposure at daily dose levels in the range of 2–50 µg/kg (Nagel *et al* 1997, vom Saal *et al* 1998, Gupta 2000). However, these results were not reproducible in two other studies, one of which included additional dose levels, and using larger group sizes compared with those used in either of the two studies showing effects (Cagen *et al* 1999, Ashby *et al* 1999).

Rubin *et al* (2001), on the other hand, reported a statistically significant and dose-dependent increase in the number of females with irregular cycles after gestational and lactational exposure to BPA at dose levels of 0.1 mg/kg bw and 1.2 mg/kg bw. Also increased body weights in males and females were seen. This effect was more pronounced at the lower dose level and in females. Salian *et al* (2009) performed a 3-generation study in Holzman strain rats and reported a significant increase in post implantation loss, decrease in litter size and decrease in sperm count and motility in the offspring of female rats dosed daily at 1.2 and 2.4 µg/kg bw of BPA by gavage. The effects were more pronounced with the positive control diethylstilbestrol (DES) at dose levels of 10 µg/kg bw. Both of these studies used, however, a limited number of parental animals and did not, for example, describe whether the litter effect was taken into account in the analysis of the results or not.

Ryan *et al* (2010) studied the effects of *in utero* and lactational exposure (from gestation day 7 to postnatal day 18) to gavaged BPA doses of 2, 20 or 200 µg/kg bw/day on sexually dimorphic behaviour, age of puberty and reproductive function of female offspring of treated rats. The results on the effects of the same BPA treatment on male offspring were published by Howdeshell *et al* 2008. No effects on female anogenital distance, pups body weights, age at vaginal opening, F1 fertility, F2 litter sizes, reproductive organ malformations, female saccharin preference and lordosis behaviour were observed. Also in males, no effects on male anogenital distance, pups body weights, androgen dependent tissue weights and epididymal sperm counts were seen.

Concerns have been raised also regarding the developmental neurotoxicity and neurobehavioural effects of BPA. Xu *et al* (2010) reported effects on memory and learning in mice exposed from gestational day 7 to postnatal day 21 to oral doses of 0.5–50 mg/kg bw. Effects were also reported e.g. by Kim *et al* (2011, postnatal exposure) at a dose level of 20 mg/kg. In some other studies, effects were seen at even lower dose levels (Miyagawa *et al* 2007, Jasarevic *et al* 2013), whereas in some other studies, no effects were observed (Jones and Watson 2012, Ferguson *et al* 2012). Stump and co-workers (2010) performed a developmental neurotoxicity study in rats according to OECD guideline 426 to address these uncertainties regarding potential neurodevelopmental effects of BPA. BPA was administered daily in the diet at concentrations of 0, 0.15, 1.5, 7.5, 75, 750 and 2 250 mg/kg feed to female Sprague-Dawley rats from gestational day 0 to postnatal day 21. Estimated intakes were 0, 0.01, 0.12, 5.85, 56.4 and 164 mg/kg bw/day during gestation and 0, 0.03, 0.25, 13.1, 129 and 410 mg/kg bw/day during lactation. The offspring were evaluated for detailed clinical observations, auditory startle, motor activity, learning and memory using the Biel water maze, brain and nervous system neuropathology and brain morphometry. No treatment related neurobehavioral effects were seen, nor was there evidence of neuropathology or effects on brain morphometry (Stump *et al* 2010). Lower body weight and body weight gain in adults and neonates were seen at the two highest dose groups resulting in a NOAEL for systemic effects of 5.85 mg/kg bw/day during pregnancy. The NOAEL for neurodevelopmental effects was the highest dose level tested. However, EFSA concluded in its evaluation 2010 (EFSA 2010) that data on Biel water maze test as performed by Stump *et al* (2010) suffer from censoring and concluded that this test on learning and memory was inconclusive and only of limited value in the risk assessment of BPA. Thus, there is still some uncertainty regarding developmental effects.

Increased or decreased anxiety (contradictory findings between the studies) has also been reported in several studies in rodents (see EFSA 2014 for a review). Because of the inconsistent findings it is not possible to conclude on these effects (EFSA 2014). Loss of sexual differences after gestational exposure has also been suggested by some studies (Carr *et al* 2003, Fujimoto *et al* 2006, Jones and Watson 2012). There are, however, a number of weaknesses in these studies including a limited number of doses and animals evaluated.

Overall, in standard reproductive and developmental studies in rodents, effects on reproduction have been seen only at high doses showing also other toxic effects. Even though several non-guideline studies suggest effects on reproductive and developmental parameters at lower dose levels (< 5 mg/kg bw), the data are contradictory and are not supported by the recent FDA/NTCR study with a wide-dose range (Delclos *et al* 2014). In humans, based on Chinese epidemiological studies, there is some concern for impaired sperm quality but, for example, the effect of other concurrent exposures cannot be excluded. In addition, there are some concerns on the potential developmental neurotoxicity of BPA based on animal studies suggesting effects on memory and learning and anxiety-like behaviour. However, since the data are very inconsistent it is difficult to conclude on the relevance of these findings.

## 4. Recommendation

### Key data

To establish a recommended occupational exposure limit (OEL), SCOEL began by considering the available data relating to inhalation exposure. In rats exposed daily to airborne BPA for 13 weeks there was a NOAEC of 10 mg/m<sup>3</sup>, with mild olfactory





epithelium inflammation at 50 and 150 mg/m<sup>3</sup>. There was no evidence of systemic toxicity in this study (Nitschke 1988).

If one considers the other toxicological evidence, most of which arises from oral studies in rodents, there were no findings that preclude the recommendation of a health-based OEL. In repeated oral dosing studies, a NOAEL of 5 mg/kg bw/day have been found for liver effects in rats and mice, with mild liver hypertrophy, increased liver weights and reductions in weight gain seen at 50 mg/kg bw/day (Stump *et al* 2010, Tyl *et al* 2002 and 2008). Also effects on kidney weight were reported in mice by Tyl *et al* (2008) at these dose levels. EFSA (2014) calculated BMDL<sub>10s</sub> for liver and kidney effects in mice, which were ~3.5 mg/kg bw/day for liver effects and 3.6 mg/kg bw/day and 3.9 mg/kg bw/day for weight changes in right and left kidney, respectively. If 100 % absorption is assumed for both exposure routes, a BMDL<sub>10</sub> of ~3.5 mg/kg bw and a NOAEL of 5 mg/kg bw at continuous subchronic exposure correspond to 34 and 49 mg/m<sup>3</sup>, respectively, at occupational inhalation exposure (8 hours/day, 5 days/week) in humans. It should be noted that these liver and kidney effects observed in oral rodent studies were very mild; even at the highest dose levels (50 and 500/600 mg/kg) only mild effects in histopathology were seen. In addition, kidney weight changes in mice at the dose levels ≤ 50 mg/kg were not accompanied by any histological changes and were not seen in rats (Tyl *et al* 2002 and 2008). The liver effects (mild hypertrophy) seen in mice, on the other hand, may have been adaptive in nature.

#### *Extrapolation issues and uncertainties in hazard assessment*

There are some species differences in the metabolism of BPA. Enterohepatic circulation in rats results in a longer half-life of BPA in rats as compared to that in humans. On the other hand, the glucuronidation rate in rats is higher than in humans. Regardless of these apparent differences in BPA toxicokinetics, levels of unconjugated BPA have been shown to remain at a very low level after oral exposure in all species (0.2–2.8 % of the total AUC, see Section 3.1).

When considering route-to route extrapolation, following oral dosing there is extensive first-pass metabolism of BPA transported directly to the liver. Following inhalation exposure, this first pass effect is missed, which may result in higher levels of free BPA after inhalation than after oral dosing. On the other hand, the maximum BPA concentration (C<sub>max</sub>) in the liver (one of the main target organs) is likely to be lower after inhalation or dermal exposure than after oral exposure. However, after inhalation of BPA it is likely that a significant part of the inhaled material becomes ingested resulting in combined oral and inhalation exposure.

There are some uncertainties related to the so-called "low-dose effects". Main concerns are related to the developmental neurotoxicity (anxiety and loss of sexual differences in behaviour) as well as possible mammary gland effects (mammary gland hyperplasia after pre- and perinatal exposure). However, there is currently no concluding evidence showing that these effects at these low levels are real, and in the latter case, relevant for human susceptibility to breast cancer. Also reports on male reproductive dysfunction in occupationally exposed persons need to be confirmed by other studies before any final conclusions can be made on the effects of BPA on human reproductive function.

#### *Overall assessment*

The inhalation NOAEC of 10 mg/m<sup>3</sup> is taken as the starting point for recommending an OEL. The critical effect in this study was respiratory tract irritation. This value of 10 is divided by an assessment factor of 3 to cover the uncertainties related to the inter-species extrapolation in these local effects resulting in a recommended OEL of 3

mg/m<sup>3</sup>. Using the preferred value approach, 3 mg/m<sup>3</sup> is rounded to 2 mg/m<sup>3</sup>. There are also some concerns about long-term systemic effects (liver and kidney effects), which may not have been fully addressed in this subchronic inhalation study. There is a 17–25-fold safety margin to the inhalation exposure levels of 34 and 49 mg/m<sup>3</sup>, which correspond to the BMDL<sub>10</sub> of ~3.5 mg/kg bw (EFSA 2014) and to the NOAEL of 5 mg/kg bw observed for kidney and liver effects in oral studies. Because liver and kidney effects observed in oral long-term rodent studies were very mild even at the highest dose levels (50 mg/kg bw and 500/600 mg/kg bw, Tyl *et al* 2002 and 2008), this margin of safety is considered sufficient to cover extrapolation to long-term exposure, and also to cover possible remaining inter- and intra-species differences in toxicokinetics and toxicodynamics.

Even though there are some concerns related to the long-term effects of BPA at exposure levels lower than 5 mg/kg bw after exposure during the foetal and early postnatal period, the results of these studies are controversial and there is no clear support for these effects at low dose levels from good quality animal studies (including the recent study by Delclos *et al* 2014). Therefore, at present SCOEL did not consider them relevant for deriving the recommended OEL.

There is no toxicological basis for recommending an additional specific short-term exposure limit (STEL).

#### *Biomonitoring*

Measurement of total urinary BPA has been used for biomonitoring of BPA exposure. In the general population, urinary BPA levels are usually below 7 µg/l (95<sup>th</sup> percentile based on German and Canadian studies). Limited data were available for recommending a BLV. Using the same formula and assumptions as used in Krishnan *et al* (2010, page 7), the recommended OEL of 2 mg/m<sup>3</sup> (meaning a daily intake of 0.29 mg/kg bw) can be calculated to correspond to a urinary level of 11.8 mg/l (13.3 mg/g creatinine) in a 70-kg male. There are, however, several uncertainties related to this calculation, the main uncertainty being related to the short half-life of BPA resulting in variation in the urinary excretion over the course of the day. In addition, the data on the toxicokinetics of BPA after inhalation or dermal exposure is limited, the majority of toxicokinetic data coming from oral exposure. Thus, no BLV can be proposed, but a biological guidance value (BGV) of 7 µg/l is recommended for the identification of potentially occupationally exposed from the occupationally non-exposed.

#### *Other assignments*

A “Sen” notation is considered not appropriate. A recent OECD guideline based study showed that skin absorption may have only a minor contribution to systemic BPA levels at the recommended OEL. Thus, no skin notation is recommended.

#### *Measurement and analysis*

Appropriate methods are available to measure airborne and urinary BPA in relation to the recommended OEL (OSHA 2013) and BGV (Fukata *et al* 2006).

*This Recommendation reflects the present knowledge on the toxicity of BPA. It will be revisited and revised when new relevant data (e.g. the results from the FDA/NCTR long-term toxicity studies) become available.*

The present Recommendation was adopted by SCOEL on 11 June 2014.

## 5. References

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